

APPENDIX C

ADDITIONAL INFORMATION ON FLUSHING IN ESTUARIES

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A variety of terms such as residence time, flushing time, transit time, turnover time, and age are used to describe time scales for transport and removal of materials that enter waterbodies. Use of these terms in the literature is often inconsistent and sometimes imprecise, so that care must be exercised to determine the meaning of terms being used, to avoid misinterpretation or incorrect comparisons of data. This appendix contains definitions of terms and discussions of factors that affect flushing and empirical and modeling methods for estimating residence times in estuaries. An understanding of residence times is especially important when estimating not only system responses to nutrients, but also the lag phase between management and system improvements.

Definitions of Residence Times

The freshwater residence time (τ_{fw}) is the average amount of time that freshwater, or a conservative tracer introduced with freshwater inputs, resides in the estuary before exiting. It is the mean transit time for a molecule of water or conservative tracer to progress from the point of input to the seaward boundary. This is the definition of residence time most often given in the literature, and is generally the most useful for analysis of eutrophication in estuaries, as most nutrients are usually introduced with freshwater.

Another commonly used concept is that of the mean amount of time required for water (or a homogeneously-distributed conservative tracer) that is in the estuary at a given time (regardless of source) to leave the estuary. This is here called “estuary residence time” (τ_e), although Takeoka (1984) and Zimmerman (1976) use the term “residence time.”

The values of τ_{fw} and τ_e may differ. Consider the case of an elongated unstratified estuary, with the seaward boundary at one end and a river as the single source of freshwater located at the opposite end. In this case, the travel distance from the river mouth to the seaward boundary is longer than the average travel distance from other parts of the estuary, so that $\tau_{fw} > \tau_e$. Conversely, if the river enters the estuary near the seaward boundary, τ_e may exceed τ_{fw} . For cases intermediate between these two, the difference between these two residence times becomes smaller, and the times may be equal. Multiple freshwater inputs and other factors such as stratification and estuary shape may complicate matters, but the same principles apply.

These concepts, the related concepts of turnover time and age, and the relationships among these measures are discussed in detail by Bolin and Rodhe (1973) and Zimmerman (1976), and summarized by Takeoka (1984). Miller and McPherson (1991) discuss another concept, “pulse residence time,” for water or a conservative constituent introduced as a pulse, that may be useful in some circumstances.

Factors That Influence Residence Time

Water residence times in estuaries are influenced by any factor that affects water movement, including freshwater inflow rates, tides, wind, mixing, stratification, and system topography. Because many of these factors are variable, residence times are also not static. This variability requires attention to the appropriate time interval over which the residence time should be expressed, and the representativeness of the conditions under which a given measurement is made. A long-term (seasonal or annual) average residence time is often most appropriate for analysis of the effects of nutrients.

Freshwater Forcing

Residence time may be quite sensitive to freshwater inflow rate, with larger flow rates associated with smaller residence times. This is illustrated by three Rhode Island estuaries. For Narragansett Bay, τ_{fw} varies between approximately 10 and 40 days; τ_{fw} at the long-term mean freshwater inflow rate is 26 days (Pilson 1985). For the Seekonk River, τ_{fw} varies between 0.4 and 2.9 days, with a value of 1.2 days for the mean freshwater inflow rate, and for the Providence River, τ_{fw} varies between 0.7 and 6.3 days, and is 2.5 days at average inflow (Asselin and Spaulding 1993).

Tidal Forcing

Tides can be a major factor controlling estuary-ocean exchange of water and therefore water residence time. Important factors are the tidal range, tidal frequency (diurnal vs. semidiurnal), and estuary depth. Tidal ranges in U.S. coastal waters range from centimeters to more than 5.5 m. For a given estuary, residence times may vary over the spring-neap tide cycle.

Wind Forcing

Wind may substantially influence estuarine circulation, and therefore water residence time. In a study of two small shallow estuaries, Geyer (1997) found that wind direction (offshore vs. onshore) had a substantial effect on salinity structure and nontidal flow in one estuary, and τ_{fw} varied by a factor of approximately 3 (from 0.8 to 2.7 days) in response to the sea-breeze cycle. Measurements in a nearby and similarly oriented estuary during the same time period found that whereas wind direction influenced salinity distributions, a constriction at the estuary mouth limited estuary-offshore exchange, so that there was no significant relationship between residence time and wind stress. Estuaries bordering the Gulf of Mexico are shallow, highly susceptible to wind forcing, and considered meteorologically dominated (Solis and Powell 1999; Ward 1980).

Determination of Residence Time

A number of empirical and computational methods are used to measure or estimate water residence times.

Empirical Measurements

Empirical measurements of water residence time depend on measurement of the distribution or dynamics of tracers (generally freshwater or introduced dye) in estuaries.

Bolin and Rodhe (1973) show that the transit time of a tracer through a reservoir is given by the turnover time (τ_o), i.e.

$$\tau_o = \frac{M_o}{F_o}$$

where M_o is the total mass of a constituent in the reservoir and F_o is the total flux through the reservoir. This is the basis for the freshwater replacement method for calculating the mean freshwater residence time in an estuary. The τ_{fw} is calculated as

$$\tau_{fw} = \frac{V_{fw}}{Q_{fw}}$$

where V_{fw} is the volume of freshwater in the estuary and Q_{fw} is the input rate of freshwater. This ratio gives the time required for the inflowing freshwater to replace the freshwater already in the estuary. The volume of freshwater in the estuary is calculated as the amount of freshwater that must be mixed with

seawater having salinity (S_s) equal to that entering at the seaward boundary to yield a volume V_e equal to that of the estuary, with salinity equal to the mean salinity of the estuary (S_e). V_{fw} may be calculated as

$$V_{fw} = \left(1 - \frac{S_e}{S_s} \right) V_e$$

Examples of this method are given by Pilson (1985), Geyer (1997), and Solis and Powell (1999).

Fluorescent dye (usually Rhodamine WT) is also used to determine water residence times. To measure freshwater residence time, dye is introduced continuously into the inflowing freshwater at a rate proportional to the freshwater flow rate. Dye concentrations are surveyed at (high or low) slack tide. Dye input is terminated when the mean concentration of dye in the estuary reaches equilibrium (approximately 3 times the freshwater residence time). Further periodic surveys are conducted at the same tide phase to monitor the dye content of the estuary. The average concentration of dye is usually found to follow a decreasing exponential with time,

$$C(t) = C_o e^{-k t}$$

where C_o is the initial average concentration of dye in the estuary and $C(t)$ is the concentration at time t . This function is fit to the data, and the mean residence time of dye in the reservoir, a surrogate for the mean freshwater residence time, is then $\tau_{fw} = 1/k$. Alternatively, as the concentration one residence time after termination of dye input is $C = C_o e^{-1}$, the time required to attain this concentration is sometimes taken as the residence time. The estuary residence time may be measured similarly. In this case, the dye is distributed as uniformly as possible throughout the estuary in a rapid application, and the change in dye content is monitored as above. The residence time determined from the rate of change of tracer concentration is sometimes called the e-folding time. Because of cost and logistical considerations, dye studies are most often done on relatively small estuaries. Examples of such studies are described by Callaway (1981), Dettmann et al. (1989), and Geyer et al. (1997).

Models

A wide range of models, ranging from simple to complex, has been used to calculate water residence time in estuaries. The simplest of these is the tidal prism model, which estimates residence time (in number of tidal cycles) as the ratio V/P , where V is the estuary volume (usually expressed at high tide) and P is the volume of the tidal prism. The model is based on the assumption that water entering the estuary on the flood tide is thoroughly mixed throughout the estuary within a tidal cycle. Most estuaries do not meet this requirement and, for all but very small estuaries, this model generally underestimates estuary and freshwater residence time, sometimes by manyfold.

Ketchum (1951) modified this simple model by dividing the estuary into segments, each having a length that corresponds to the local tidal excursion. Water exchanges between adjacent segments during each tidal cycle are calculated, and complete mixing is assumed to occur only within each segment. The segmented tidal prism model requires data for freshwater inputs, tidal range, and estuary topography. The simple and segmented versions of the tidal prism model are reviewed by Pritchard (1952) and Dyer (1973), and modified by Dyer and Taylor (1973). For most (but not all) estuaries to which the segmented model has been applied, it has given good results (Dyer 1973). Beyond permitting calculation of residence time, segmented tidal prism models permit calculation of the distributions of fresh- and

saltwater and other water quality constituents along the estuary, but do not address the effects of wind on flushing.

Another approach to calculating residence times is the box model. Box models also segment the estuary, assume complete mixing within segments, and calculate diffusive and advective exchanges between adjacent segments. These models require data for estuary topography and for freshwater inflows and salinity distribution in the estuary for each set of conditions to which the model is applied. As actual salinity data are used, the effects of wind and tide are implicitly taken into account. Applications of such models are described by Brown and Arellano (1980), Dettmann et al. (1992), Hagy et al. (2000), and Miller and McPherson (1991). Such models also allow calculation of advective and diffusive exchanges among segments and spatial distribution of water quality constituents.

Numerical computer models of hydrodynamics and constituent transport are still more complex. These models are used to simulate movement of water and water quality constituents at fine spatial resolution, and may also be used to calculate freshwater or estuary residence times. These models generally allow consideration of wind stress as well as tidal and freshwater forcing. A few examples of such models are described by Brooks et al. (1999), Signell (1992), and Signell and Butman (1992).

APPENDIX D
NOAA SCHEME FOR DETERMINING ESTUARINE SUSCEPTIBILITY
(SOURCE: BRICKER ET AL. 1999)

The following provides an overview of the NOAA scheme for determining the overall human potential for causing nutrient enrichment of estuaries.

Dilution and Flushing Potential

The length of time that nutrients spend in an estuary potentially affects their opportunity to contribute to overenrichment. The time is a function of dilution potential and flushing rate. The analysis uses physical and hydrologic data to define separately (1) a dilution rating and (2) a flushing rating. In both cases, the higher rating, the greater the capacity to dilute or flush nutrient loads (see Tables D-1 and D-2).

Figure D-1 combines dilution potential and flushing potential. By combining dilution and flushing components, an export potential (EXP) is determined. Estuaries in the upper left portion of the matrix generally have a high EXP that suggests an ability to dilute and flush nutrient loads. Estuaries in the lower right portion of the matrix have the opposite capacity, making them more susceptible to nutrient input.

Nutrient Inputs

NOAA used the USGS Sparrow Model (spatially referenced regressions of contaminant transport on watershed attributes) and other information to estimate nutrient loads and measure nitrogen pressure on estuaries.

Determination of Overall Human-based Nutrient Pressure

A matrix was used to compare the susceptibility to nutrient retention and the level of N inputs to rank the overall expression of human influence on eutrophic conditions in the estuary (Figure D-1). Experts at the National Assessment Workshop reviewed and, where appropriate, modified the assessments based on higher quality data available for some estuaries; expert knowledge also played a role.

Table D-1. Decision rules for dilution potential

Type	IF: vertical stratification	THEN: dilution volume	IF: dilution value	Dilution potential	No. estuaries
A	Vertically homogenous •all year •throughout estuary	$1/VOL_{\text{estuary}}$	10^{-13} 10^{-12}	High	30
B	Minor vertical stratification •navigation channels •upper estuary	$1/VOL_{\text{estuary}}$	10^{-11}	Moderate	63
C	Vertically stratified •most of year •most of estuary	$1/VOL_{\text{fwf}}$ (fwf=freshwater fraction)	10^{-10} 10^{-09}	Low	45

Note: This analysis assumes that a larger portion of the water column is potentially available to dilute nutrient loads in a vertically homogeneous estuary than in a vertically stratified system. The assumption is that for stratified systems, nutrients are most often retained in the upper portion (freshwater fraction) of the water column. In contrast, downward transport (more complete mixing) is likely in vertically homogeneous systems. Type B estuaries are generally vertically homogeneous, although stratification is observed (confined) in narrow navigation channels or the extreme upper reaches of an estuary. In this case, nutrients are assumed to be diluted throughout the entire water column.

Source: Bricker et al. 1999.

Table D-2. Decision rules for flushing potential

Type	Tide range (ft)	Freshwater inflow/estuary volume	Flushing potential	No. estuaries
1	Macro (>6)	and Large or moderate (10^{00} to 10^{-02})	High	12
2	Macro (>6)	and Small (10^{-03} , 10^{-04})	Moderate	21
3	Meso (>2.5)	and Large (10^{00} , 10^{-01})	High	15
4	Meso (>2.5)	and Moderate (10^{-02})	Moderate	16
5	Meso (>2.5)	and Small (10^{-03} , 10^{-04})	Low	26
6	Micro (<2.5)	and Large (10^{00} , 10^{-01})	High	4
7	Micro (<2.5)	and Moderate (10^{-02})	Moderate	13
8	Micro (<2.5)	and Small (10^{-03} , 10^{-04})	Low	31

Note: This analysis assumes that a greater capacity to flush nutrient loads exists for estuaries that have large tide and freshwater influences.

Source: Bricker et al. 1999.

ESTUARINE EXPORT POTENTIAL AND SUSCEPTIBILITY

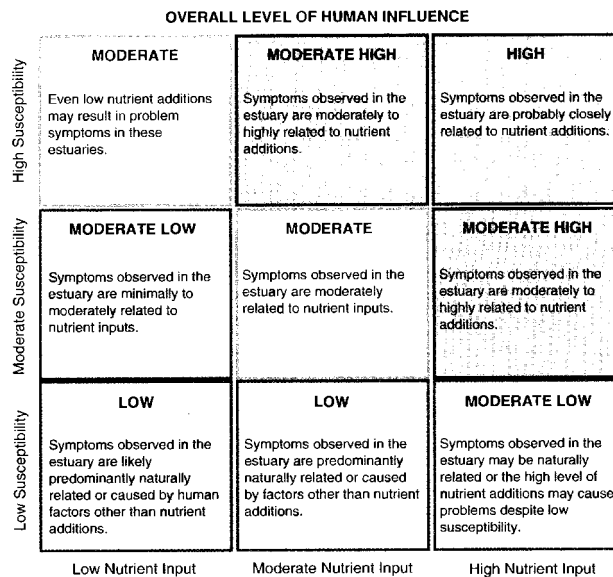
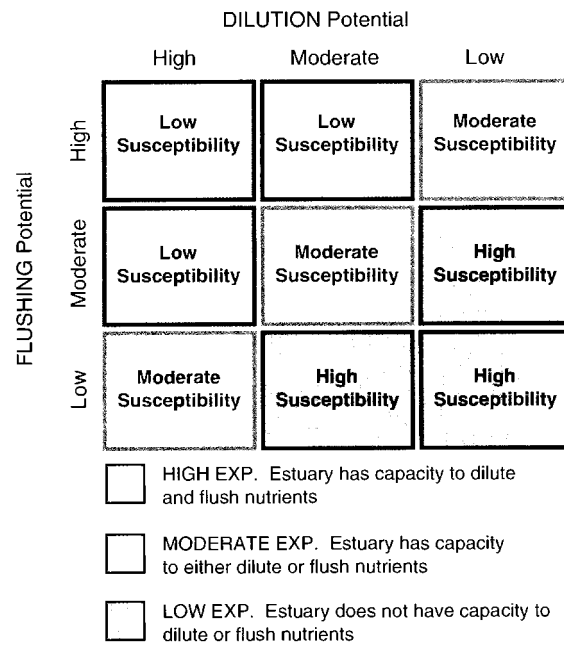


Figure D-1. Estuarine Export Potential Susceptibility.

APPENDIX E

COMPARATIVE SYSTEMS EMPIRICAL MODELING APPROACH: THE EMPIRICAL REGRESSION METHOD TO DETERMINE NUTRIENT LOAD-ECOLOGICAL RESPONSE RELATIONSHIPS FOR ESTUARINE AND COASTAL WATERS

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The empirical regression method can be used to determine the response of estuarine systems to nutrient loading. This method can take two forms: single system and comparative-systems approaches.

In single system approaches, ecological responses and nutrient concentrations are measured over time in a single system. This allows for the development of load-response models for the individual systems that are robust because many of the controlling factors, such as, for example, physiographic setting, primary productivity base, and hypsography, are relatively constant. This approach has been used to develop models of the response of primary productivity to nitrogen load (Figures E-1a,b) (Boynton et al. 1995). However, this approach is only applicable to the system in which it was developed and thus is not considered to be very useful in the development of widely applicable nutrient load-response relationships. Single system empirical studies, however, may be useful in providing data on processes useful for numerical model development.

The alternative approach, using the space for time paradigm (Pickett, 1988), posits that relationships between nutrient inputs and ecologically meaningful estuarine responses, using multiple systems, have predictive capability, at least for the category of systems used in the model development. This allows for a wide range in nutrient loading and estuarine types to be included. The comparative-systems empirical approach has been used to determine, for example, relationships between nutrient inputs and fish yields (Lee and Jones 1981; Nixon 1992), benthic biomass, production and abundances (Josefson and Rasmussen 2000), summer ammonia flux (Boynton *et al.*, 1995), chlorophyll-a concentration (Boynton et al. 1996; Boynton and Kemp 2000; Monbet 1992), primary productivity (Nixon *et al.*, 1996), and the dominant source of primary productivity (Nixon et al. in press). In many of these cases, important environmental factors such as flushing time and depth, are used to normalize the nutrient loading in a similar way as Vollenweider (Vollenweider 1976) to yield more precise relationships.

The comparative-systems empirical approach has been successfully used in a regulatory framework to develop total maximum loads for 30 subestuaries within Buzzards Bay in Massachusetts (Costa et al. 1999). Using a citizens' monitoring network, many important water quality variables were measured during summer sampling periods over a five to seven year period. Nitrogen inputs were estimated from a land-use model modified from Valiela (Valiela et al., 1997) augmented with literature data on point source and atmospheric inputs. In this study, nitrogen load-response relationships were derived for nitrogen concentration, chlorophyll-a, secchi depth, dissolved oxygen concentration, and eelgrass habitat ratio. Nitrogen loadings were normalized to account for volume and flushing time of each of the systems to improve the precision of the empirical models. An example of the types of relationships determined using this approach are given in Figure E-2. The regional estuarine management program used this method to adopt total maximum annual loads (TMALs) for nitrogen. Specifically, "...the nitrogen management strategy represents a linking of estimates of watershed nitrogen loading...to empirical observations of ecosystem response in a wide variety of Buzzards Bay embayments" (Costa 2000).

The comparative-systems empirical approach does not explicitly consider the processes that produce the observed phenomena; however, factors known to affect behavior are determined in order to reduce the

uncertainty in the models. The smaller the variance in the load-response relationship, the more compelling the association. One important question for management is to determine what level of variance is sufficient to convince stakeholders to accept nutrient limits and associated monetary expenditures.

This approach is largely based on statistical associations and is therefore restricted to prediction within the class of systems used in the model development. Applicability improves, however, by the inclusion of systems that encompass a wide range in loading and responses. The comparative-systems empirical approach allows for the direct measurement of important endpoints (e.g., hypoxia, SAV loss, biomass) obtained in the environment. The endpoints are what are important to the general public. So by providing a mathematical relationships between the stressor and important endpoints, managers can convince the public of the importance of regulatory action.

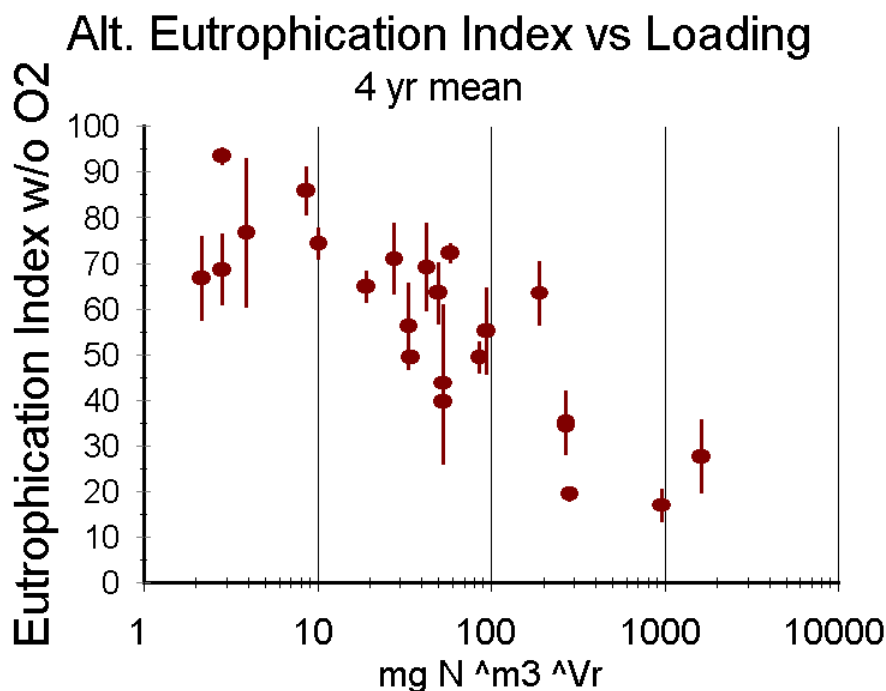


Figure E-1a. Scatter plots showing correlation between nitrogen loading, expressed using the volume Vollenweider-term flushing scale, and 92-98 mean +/- std. errors of the Alternate Eutrophication Index scoring (without oxygen scores).

Eelgrass habitat cover vs. Loadings

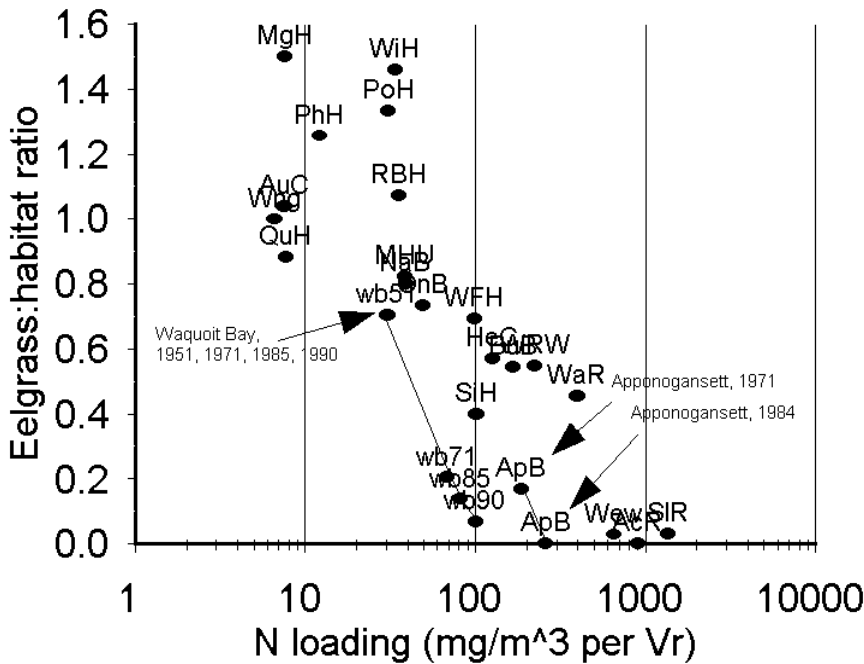


Figure E-1b. Ratio of eelgrass habitat area to potential habitat area versus nitrogen loading, expressed using the volume Vollenweider-term flushing scale (from Costa et al., 1999).

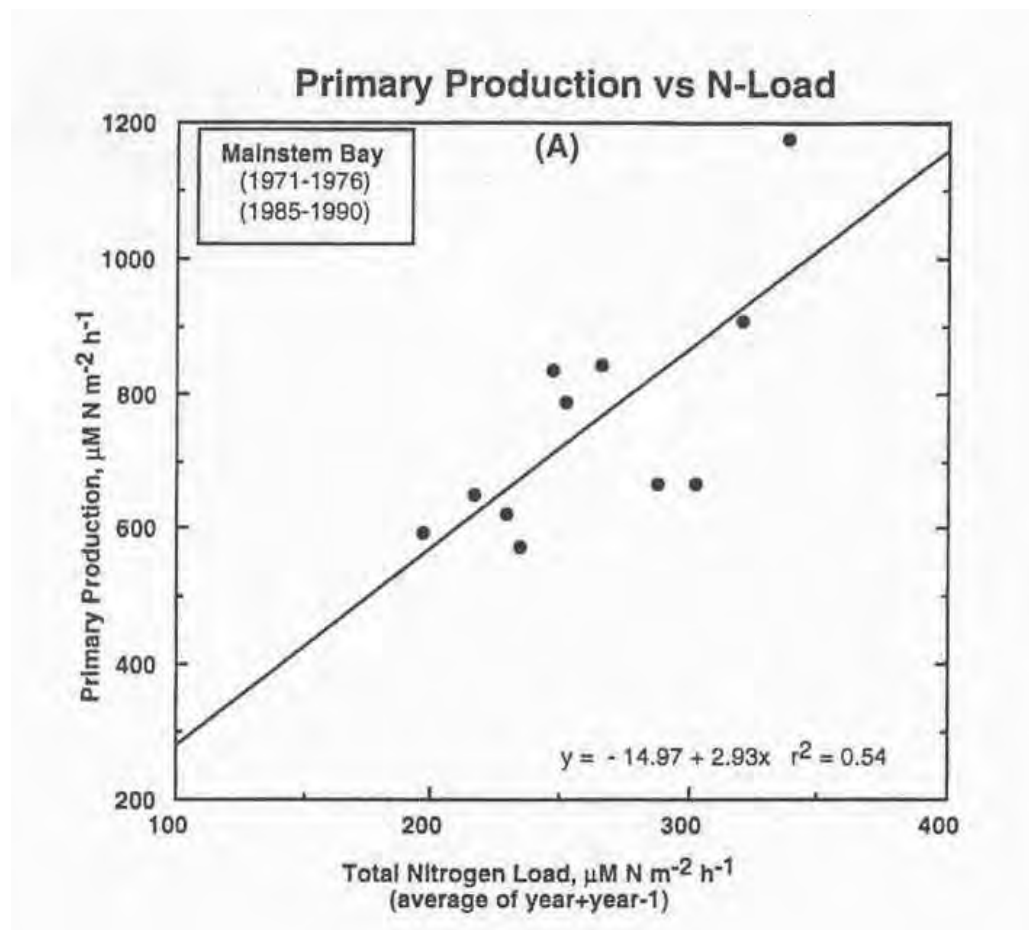


Figure E-2. Plot of annual TN loading rates versus phytoplankton primary production rates at a single station in the Chesapeake Bay from 1971-76 and 1985-90. Source: Boynton et al., 1995.

APPENDIX F

SELECTED THEORETICAL APPROACHES TO CLASSIFICATION OF ESTUARIES AND COASTAL WATERS

Several theoretical schemes have been presented that may facilitate classification of estuaries and coastal waters if the need arises for more theoretical approaches. In most cases, these schemes have potential future value but are not likely to be immediately useful. The information-thermodynamic approach offered by Ulanowicz would seem to provide useful insights to development of an eutrophication index and may be worth an earlier consideration.

Functional Attributes

Odum and Copeland (1974) proposed a classification scheme based on the idea that an ecosystem is a balance between energies that build structure and order, or *ordering energies*, and energies that cause loss of structure and order, or *disordering energies*. Although this approach is of theoretical interest, the data to apply it are still largely unavailable. There are conceptual difficulties. Energy sources and stresses are not always mutually exclusive categories. Day et al. (1989) give several examples of cases where moving water can be either an energy source or a stress, depending on the situation. Moderate currents are a source of energy for seagrass meadows, because they transport organic matter and inorganic nutrients to beds and remove metabolic wastes. If currents are too strong, however, they can disrupt beds and act as disordering energies.

Theoretic Indices

This approach is data intensive and has potential as appropriate estuary data are generated. Ulanowicz (1986, 1997) described an approach based on flow analysis and information theory. The idea is that an ecosystem can be characterized in terms of growth and organization. Growth is defined as an increase in system activity or total system throughput (analogous to total system energy flow). Organization is equated with the mutual information inherent in the trophic flow structure. To apply this approach, one would need to obtain energy flow values (or organic carbon-based equivalents) among trophic compartments. Following Ulanowicz's formulation, one could develop an information index of eutrophication. Compiling such would entail an assessment of the growth and organization status of a eutrophic or nutrient-enriched system compared with a current reference system, a minimally impaired system, or an estimate of pre-eutrophication values from historical data. The ratio of growth and organization of a nutrient-enriched system to its reference value would express the degree of impairment. Such information could help guide restoration priorities. The downside is that the approach requires the availability of a rich ecological database.